Metals

Ecological Toxicity Methods and Metals

An examination of two case studies

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Abstract

Goal, Scope and Background. The Apeldoorn Workshop (April 15th, 2004, Apeldoorn, NL) brought together specialists in LCA and Risk Assessment to discuss current practices and complications of the life cycle impact assessment (LCIA) ecological toxicity (ecotox) methodologies for metals. The consensus was that the LCIA methods currently available do not appropriately characterize impacts of metals due to lack of fundamental metals chemistry in the models. A review of five methods available to perform ecotox impact assessment for metals has been prepared to provide Life Cycle Assessment (LCA) practitioners with a better understanding of the current state of the science and potential biases related to metals. The intent is to provide awareness on issues related to ecotox impact assessment.

Methods. In this paper two case studies, one a copper based product (copper tube), the other a zinc-based product (gutter systems), were selected and examined by applying freshwater ecological toxicity impact models – USES-LCA, Eco-indicator 99 (EI 99), IMPACT 2002, EDIP 97, and CalTOX-ETP. Both studies are recent, comprehensive, cradle-to-gate, and peer-reviewed. The objective is to review the LCIA results in the context of the practical concerns identified by the Apeldoorn Declaration, in particular illustrating any inconsistencies such as chemical characterization coverage, species specificity, and relative contribution to impact results.

Results and Discussion. The results obtained from all five of the LCIA methods for the copper tube LCI pointed to the same substance as being the most important – copper. This result was obtained despite major fundamental differences between the LCIA methods applied. However, variations of results were found when examining the freshwater ecological toxicity potential of zinc gutter systems. Procedural difficulties and inconsistencies were observed. In part this was due to basic differences in model nomenclature and differences in coverage (IMPACT 2002+ and EDIP 97 contained characterization factors for aluminium that resulted in 90% and 22% contribution to burden respectively, the other three methods did not). Differences were also observed relative to the emissions source compartment. In the case of zinc, air emissions were found to be substantial for some ecotox models, whereas, water emissions results were found to be of issue for others.

Conclusions. This investigation illustrates the need to proceed with caution when applying LCIA ecotox methodologies to life cycle studies that include metals. Until further improvements are made, the deficiencies should be clearly communicated as part of LCIA reporting. Business or policy decisions should not without further discussion be based solely on the results of the currently available methods for assessing ecotoxicity in LCIA.

Outlook. The outlook to remedy deficiencies in the ecological toxicity methods is promising. Recently, the LCIA Toxic Impacts Task Force of the UNEP/SETAC Life Cycle Initiative has formed a subgroup to address specific issues and guide the work towards establishment of sound characterization factors for metals. Although some measure of precision of estimation of potential impact has been observed, such as in the case of copper, accuracy is also a major concern and should be addressed. Further investigation through controlled experimentation is needed, particularly LCIs composed of a variety of inorganics as well as organics constituents. Support for this activity has come from the scientific community and industry as well. Broader aspects of structure and nomenclature are being collectively addressed by the UNEP/SETAC Life Cycle Initiative. These efforts will bring practical solutions to issues of naming conventions and LCI to LCIA flow assignments.

Keywords: Apeldoorn Declaration; life cycle impact assessment (LCIA); life cycle management (LCM); metals; toxicity methods

1 Goal and Scope

The Apeldoorn Workshop (April 15th, 2004, Apeldoorn, NL) brought together specialists in LCA and Risk Assessment to discuss current practices and complications of the life cycle impact assessment (LCIA) ecological toxicity methodologies for metals (Ligthart et al. 2004). The consensus of the workshop was that LCIA ecotox methods currently available do not appropriately characterize potential impacts of metals due to lack of fundamental metals chemistry in the models. This resulted in a Declaration calling for: clear communication of the deficiencies of present characterization models in LCIA reporting; improved models and characterization factors that would include metals species (preferably in terms of dissolved metal instead of total metal); and

sensitivity analysis to be made when contribution analysis demonstrates that metals have a dominant influence on results and conclusions.

A review of five methods to perform life cycle ecological toxicity (ecotox) impact assessment for metals has been prepared to provide Life Cycle Assessment (LCA) practitioners and commissioners with a better understanding of the current state of the science. For clarity, this investigation was limited to five freshwater ecological toxicity models. They are: (1) the Uniform System for the Evaluation of Substances (USES)-LCA model (Huijbregts et al. 2000a, 2000b), in the Dutch Handbook method (Guinée et al. 2002); (2) the IMPACT 2002 model (Pennington et al. 2005)[1], in the IMPACT 2002+ method (Jolliet et al. 2003); (3) the ecological toxicity model in the Environmental Design of Industrial Products (EDIP) 97 method (Hauschild and Wenzel 1998), (4) the CalTOX Ecological Toxicity Potential (CalTOX-ETP) model (McKone 2001) in the US Environmental Protection Agency's Tool for the Reduction and Assessment of Chemical and other environmental Impacts (TRACI) method (Bare et al. 2003); and (5) the Ecosystem Quality model which in the Eco-indicator 99 (EI 99) method developed by PRé Consultants (Goedkoop and Spriensma 1999).

The objectives of this paper are: to inform LCA practitioners with a better understanding of how LCIA methods assess the impacts of metals; to illustrate metal-specific LCIA issues observed in two case studies (copper and zinc); to provide awareness for LCA practitioners of potential bias due to metals-specific issues. The paper does not compare the relative human health and environmental performance of copper and zinc; provide information to support decision-making directly related to the case studies presented; nor provide a description of a new method or model and improved ecological toxicity impact characterization factors. Rather this paper is an investigation of five freshwater ecological toxicity methods from a practitioner's perspective, and whether the current state-of-the-science can support practical decision-making in a life cycle management context.

2 Background

The overarching tenets of LCA are rooted in common sense – to ensure truly preferable options that lessen the burden are selected and options that merely shift burden, or even worse, have the potential to increase burden, are discounted. To this point of common sense, Heijungs et al. (2004) present a troublesome example illustrating that current ecotox models suggest that it is better to empty a jerrycan of benzene into a fountain than to throw a nickel coin into it.[2]

Procedurally, LCIA is a ranking of the results of the life cycle inventory (LCI) to quantify potential environmental harm. As depicted in **Fig. 1**, LCIA creates the connection between environmental interventions (i.e. the inventory) and damage classes of endpoints, referred to as areas of protection (AoPs) (Udo de Haes and Lindeijer 2002).

In this context, two approaches to LCIA are distinguished: a midpoint approach and an endpoint approach. The midpoint approach starts from the environmental interventions that bring us further along the environmental mechanism of impact categories (typically a subset of those presented in Fig. 1) (Jolliet et al. 2004). In contrast, the endpoint approach starts at the endpoint of environmental mechanisms, where damage occurs. The critical issues of concern by society are chosen at this endpoint level (Udo de Haes and Lindeijer 2002). Of the five freshwater ecological toxicity models examined in this paper, four include midpoint characterization models: USES-LCA, IMPACT 2002+, EDIP 97, and CalTOX-ETP, with the ecological toxicity model of EI 99 classified as a damage assessment model.

[1] The specific reference Pennington et al. (2005) of the ecological toxicity method IMPACT 2002 developed for IMPACT 2002+, in Jolliet et al. (2003) was not available at the time of publication.

[2] The USES-LCA equivalents calculate a 0.45 kg 1,4-DCB equivalent for 5 kg of benzene versus 32 kg 1,4-DCB equivalent for a 10g nickel coin (Guinée et al. 2002).

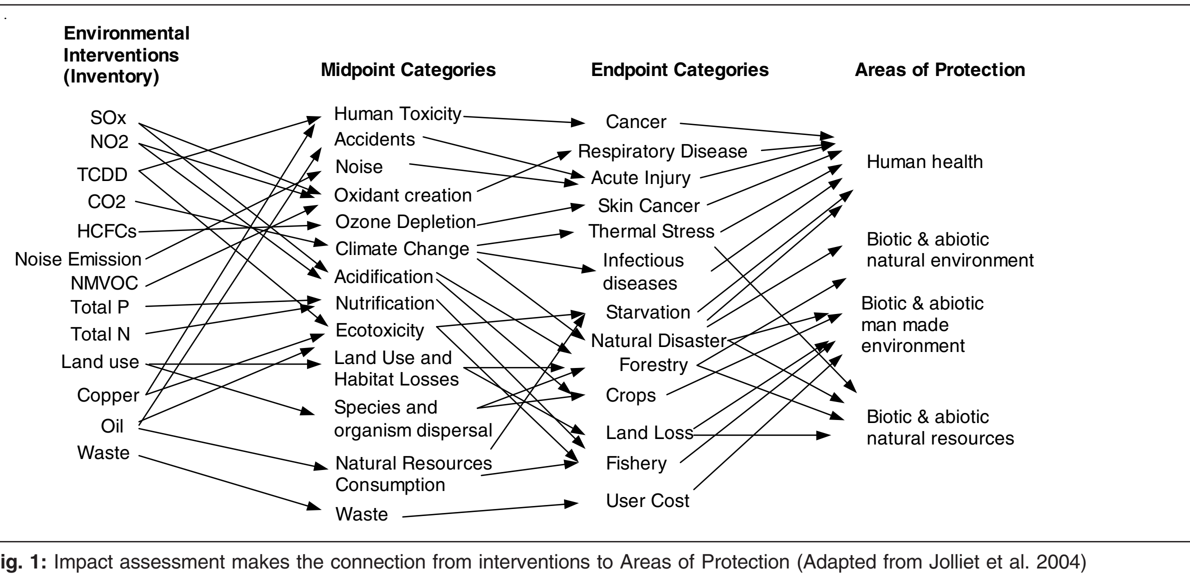

Fig. 1: Impact assessment makes the connection from interventions to Areas of Protection (Adapted from Jolliet et al. 2004)

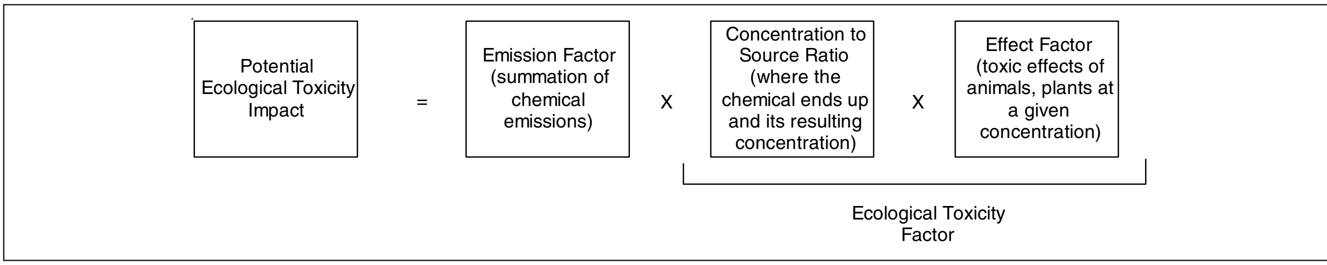

Fig. 2: Ecological Toxicity – General Framework (Jolliet et al. 1996)

2.1 Ecological toxicity modeling general framework

The framework for the five ecological toxicity models examined is generally based on three factors (the EI 99 method includes an additional damage factor). These are (1) an emissions factor (summation of chemical emissions) (2) a concentration-to-source (CSR) factor (where the chemicals end up and their ultimate concentration), and (3) a toxicity effect factor (toxic effects to animals, plants at a given concentration) (**Fig. 2**).

The emission factor is the summation of emissions and flow definitions. Flow definitions indicate precisely which impact category is related to the emission of a chemical. The product of the CSR factor and effect factor represent the ecological toxicity factor (the characterization factor). There is a unique ecological toxicity factor for every chemical identified by an impact assessment model. The most frequently used effect factor is the Predicted No-Effect Concentration (PNEC) indicator (USES-LCA, CalTOX-ETP, EI 99, and EDIP 97). The PNEC represents the highest concentration to cause no effects to the most sensitive species (EC 1996). Recent advancements have incorporated a hazard concentration affecting 50% of the species (Impact 2002 model). The basis for improvement is founded on the fundamental aspect of LCA as comparative tool, rather than for determining absolute risk. Using direct measures of toxicity on a set of representative species is more certain than interpolated measures on the most sensitive species (Jolliet et al. 2003).

2.2 Fundamental issues

In spite of recent advances, fundamental issues remain regarding how the ecotox models handle bioavailability, speciation, persistence and the essentiality of metals (Ligthart et al. 2004).

The current ecological toxicity impact assessment models are based on relating total concentration to potential effect or impact. However, it is known that toxicity is not simply related to total dissolved metal concentrations (Di Toro et al. 2001). Toxicity can also be characterized by concentration thresholds at receptor sites, such as the gills of fish (Meyer et al. 1999). Further, formation of organic and inorganic metal species can render a significant fraction of dissolved metals non-bioavailable. For example, the toxicity of a metal speciated by natural complexing agents, such as carbonates, can be one tenth the toxicity of 'free' or inorganic metal (Morel et al. 1993). In this sense, LCI measures of total elemental metals can be over conservative and misleading (Heijungs et al. 2004).

In additional, issues of persistence have also been observed. For example, Huijbregts et al. (2001) compared the toxicity potentials calculated by USES-LCA for time horizons 20, 100 and 500 years to toxicity potentials with for an infinite time horizon. For persistent organic substances, such as endrin, the relative difference between toxicity potentials calculated for an infinite time horizon and a time horizon of 20 years were limited to 0.5 order of magnitude. In contrast, time horizon dependent differences for metal toxicity potentials were found to be several orders of magnitude suggesting ecological impacts could be delayed and continue to occur over an extended period of time.

An issue of lesser concern is disregard for the essentiality of metals. The current practice of LCIA is based on the paradigm of 'less is better'.[3] However, a number of metals are essential for life functions (e.g., iron for oxygen transport, zinc for gene expression, copper for bone elasticity). Each plant and animal species has an optimal range of essential metal concentration, however, current practice ignores potential impacts due to deficiency.

3 Methods

In this paper two LCA case studies, one a predominately copper based product, the other a predominately zinc-based product, are examined by applying the following publicly available freshwater ecological impact assessment methods – USES-LCA, EI 99, IMPACT 2002, EDIP 97, and CalTOX-ETP. Both studies are recent, comprehensive, peer-reviewed case studies, completed before the Apeldoorn Workshop. The objective is to review the LCIA results in the context of the practical concerns identified by the Apeldoorn Declaration. In particular, this investigation will document the occurrence of any procedural difficulties or inconsistencies that may be observed. For the purposes of clarity of investigation and presentation of results, the LCIs investigated are limited to their cradle-to-gate results.

3.1 Cast study 1: Cradle-to-gate LCA of copper tube

The International Copper Association (ICA) commissioned a LCA study of copper tube used in residential housing (Ecobalance 2000a, 2000b). The study looked at using cop-

[3] There are exceptions, e.g., the photo oxidant chemical potential (POCP) characterization factor for nitric oxide (NO) is negative in The Dutch Handbook method (Guinée et al. 2002). This implies an unbounded 'more is better' paradigm.

per tube to deliver water within a typical American house over the life time of the house. An important aspect of the study was the collection of data from copper production sites in the United States. Collection of this data provided a comprehensive and up-to-date life cycle inventory for the copper industry. Data was received from thirty-one sites, representing 58% of the copper production by refinery and 74% by solvent extraction and an external critical review was conducted. The boundary includes the major process steps for copper tube production: mining, milling, smelting, refining, solvent extraction/electrowinning (SX/EW) and tube manufacturing.

3.2 Case study 2: Cradle-to-gate LCA of a zinc gutter systems

TNO Environment, Energy and Process Innovation (TNO-MEP) was commissioned by three manufacturers (NedZink B.V. (Netherlands), Rheinzink GmbH & Co. KG (Germany) and Union Minière s.a. (Belgium and France)) to carry out a LCA study representative of zinc gutters and down pipes installed and used in the Netherlands (Eggels et al. 2000). Data from the three manufacturers represent 90% of the Dutch market. Rainwater drainage by gutters is mainly used in residential market applications and consists of several material choices: aluminium, concrete, copper, glassfiber reinforced polyester, poly vinyl chloride (PVC), glassfiber reinforced PVC, steel, and zinc (zinc-copper-titanium alloy). Zinc is the material of choice in 70% of the gutter applications, PVC is ranked second.

This study includes a life-cycle assessment (LCA) for the manufacture of a zinc gutter by the three commissioners. The study was conducted within the framework of Milieu-relevante Productinformatie (MRPI). The functional unit of the study is represented by an average zinc gutter system for a residential home. Data related to production of the zinc gutter was based on the market share of the three manufacturers. Accessories that are required for the gutter to fulfil the function have been incorporated into the functional unit. These include four end pieces of the same zinc-copper-titanium alloy and sixteen supporting brackets (galvanized steel with a zinc layer of 30 µm on both sides). The main raw material for the production of zinc gutters is special high grade (SHG) zinc, which has a purity of 99.995%. The source of the process data is from a confidential report, Boustead (1998), commissioned by the International Zinc Association (IZA). The data in the report covers 75% of the Western-European zinc ingot/alloy production.

4 Results

4.1 Results of copper tube investigation

The LCI output results for emissions to air and water for the production of 1 kg of copper tube are shown below in **Table 1**. The contribution to emissions for the three main processes – hydrometallurgically (hydro) produced cathode copper, pyrometallurgically (pyro) produced cathode copper and the production of copper tube are presented in detail.

Table 1: LCI output results for 1 kg of copper tube with contribution by process stage

Flow [(a) air; (w) water]	Units	Total Emission
(a) Ammonia (NH_3)	kg	5.15E-03
(a) Arsenic (As)	kg	6.06E-06
(a) Cadmium (Cd)	kg	2.26E-06
(a) Carbon Dioxide (CO_2, fossil)	kg	3.69E+00
(a) Carbon Monoxide (CO)	kg	1.00E-02
(a) Chlorides (Cl-)	kg	3.58E-07
(a) Chromium (Cr III, Cr VI)	kg	2.15E-06
(a) Cobalt (Co)	kg	1.32E-07
(a) Copper (Cu)	kg	1.23E-04
(a) Hydrocarbons (except methane)	kg	1.58E-03
(a) Hydrocarbons (unspecified)	kg	2.81 E-04
(a) Lead (Pb)	kg	2.05E-05
(a) Mercury (Hg)	kg	5.09E-08
(a) Metals (unspecified)	kg	1.08E-05
(a) Methane (CH_4)	kg	9.49E-03
(a) Nickel (Ni)	kg	1.39E-06
(a) Nitrogen Oxides (NO_x as NO_2)	kg	1.85E-02
(a) Nitrous Oxide (N_2O)	kg	3.64E-04
(a) Particulates (PM 10)	kg	2.34E-03
(a) Particulates (unspecified)	kg	1.35E-02
(a) Silver (Ag)	kg	1.07E-07
(a) Sulfur Oxides (SO_x as SO_2)	kg	1.52E-02
(a) Sulfuric Acid (H_2SO_4)	kg	6.02E-03
(a) VOC (Volatile Organic Compounds)	kg	6.06E-04
(a) Zinc (Zn)	kg	1.07E-05
(w) Acids (H+)	kg	6.22E-06
(w) BOD5 (Biochemical Oxygen Demand)	kg	2.21 E-04
(w) Cadmium (Cd++)	kg	4.64E-09
(w) Chromium (Cr III, Cr VI)	kg	2.76E-08
(w) Cobalt (Co I, Co II, Co III)	kg	9.70E-11
(w) COD (Chemical Oxygen Demand)	kg	9.52E-04
(w) Copper (Cu+, Cu++)	kg	1.96E-06
(w) Cyanide (CN-)	kg	1.86E-08
(w) Lead (Pb++, Pb_4+)	kg	1.25E-07
(w) Mercury (Hg+, Hg++)	kg	3.44E-10
(w) Nickel (Ni++, Ni_3+)	kg	6.93E-07
(w) Silver (Ag+)	kg	3.64E-12
(w) Sulfate (SO_4--)	kg	1.41E-04
(w) Suspended Matter (unspecified)	kg	4. 14E-04
(w) TOC (Total Organic Carbon)	kg	5.04E-07
(w) Zinc (Zn++)	kg	8.45E-07

The results of the LCI show that the major air and water emissions by mass are non-metals. Emissions data reported for metals are generally specified as elemental species for air emissions and as cation species for water releases. The LCI data assembled in Table 1 were then classified and applied

Table 2: Copper tube freshwater ecotox results by percent contribution

Flow [(a) air; (w) water]	USES-LCA FAETP	CalTOX ETP	EDIP 97 EcoTox	IMPACT 2002+	EI99 HA EcoTox
(a) Arsenic (As V) or (As)	0.88%	0.16%	0.58%	0.08%	1.17%
(a) Cadmium (Cd++) or (Cd)	1.92%	0.04%	13.24%	0.24%	7.10%
(a) Chromium (Cr VI) or (Cr)	0.01%	0.04%	0.07%	0.04%	2.89%
(a) Copper (Cu++) or (Cu)	80.08%	96.53%	76.54%	88.45%	58.46%
(a) Lead (Pb)	0.14%	0.02%	2.04%	0.20%	16.95%
(a) Mercury (Hg++) or (Hg)	0.05%	0.16%	0.05%	0.01%	0.01%
(a) Nickel (Ni++) or (Ni)	2.57%	0.79%	0.05%	0.06%	3.21%
(a) Zinc (Zn++) or (Zn)	0.56%	1.77%	0.53%	0.53%	10.07%
(w) Copper (Cu++) or (Cu)	6.66%	0.43%	6.10%	9.85%	0.09%
(w) Nickel (Ni++) or (Ni)	6.59%	0.01%	0.11%	0.22%	0.03%
Total Contribution	**99.47%**	**99.95%**	**99.31%**	**99.67%**	**99.99%**

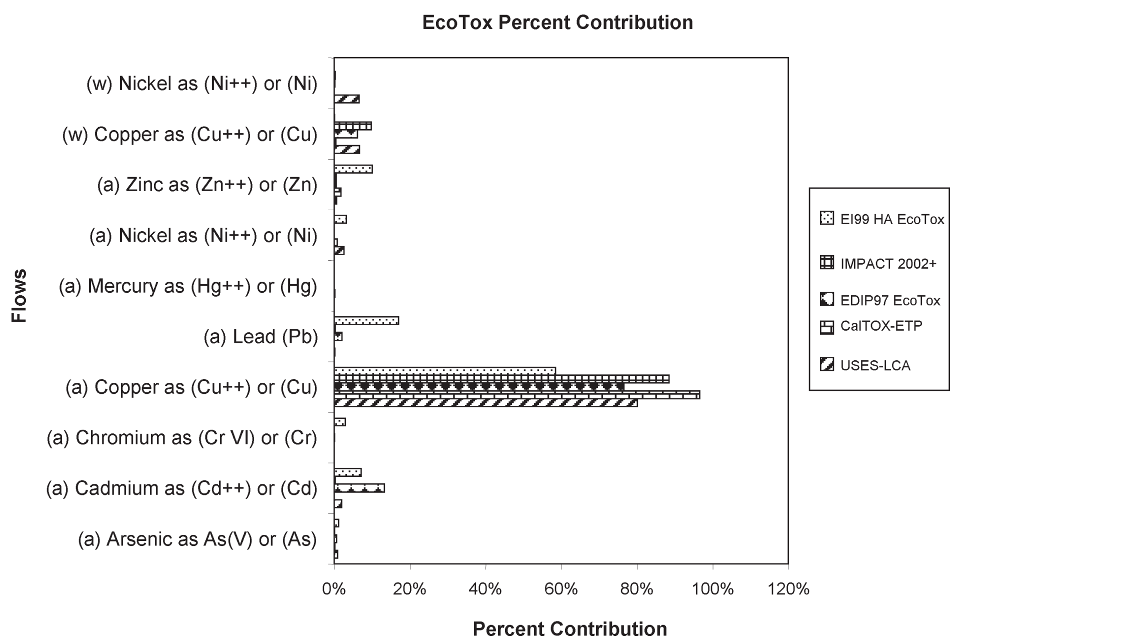

Fig. 3: Copper tube freshwater ecotox results by percent contribution

to the five ecotox models. The detailed classification and characterization results are presented in **Table 6** (**see Appendix,** OnlineFirst, http://dx.doi.org/10.1065/lca2006.01.229). The contribution analysis of the LCIA results are summarized in **Table 2** and depicted in **Fig. 3**. Chemical emissions that exceeded 1% contribution to total impact for any impact method are shown.

4.2 Results of zinc gutter system investigation

Results of the output emissions to air and water for the zinc gutter LCI are presented in **Table 3**. The final results are presented based on 'average gutter' production for a reference house. Emissions are shown for the three main processes – primary zinc production, steel production and the gutter system production.

Similar to copper tube production, the results of the zinc LCI show that the major air and water emissions by mass are non-metals and the emissions data reported for metals were generally specified as elemental species for air emissions and cation species for water releases. The zinc LCI data assembled in Table 3 were then classified and applied to the five freshwater ecotox models. The detailed classification and characterization results are presented in **Table 7** (**see Appendix,** OnlineFirst, http://dx.doi.org/10.1065/lca2006.01.229). The contribution analysis of the LCIA results are summarized in **Table 4** and depicted in **Fig. 4**. Chemical emissions that exceeded 1% contribution to total impact for any impact model are shown.

5 Discussion

The results of the examination of the two cases were mixed. On the one hand, the results obtained from all of the LCIA methods for the copper tube LCA pointed to the same substance as being the most important – copper. This result was

Table 3: LCI output results for zinc gutter system (Eggels et al. 2000)

Flow [(a) air; (w) water]	Units	Gutter Route
(a) Ammonia (NH_3)	kg	1.41E-02
(a) Carbon Dioxide (CO_2, fossil)	kg	8.70E+01
(a) Carbon Monoxide (CO)	kg	1.40E-01
(a) Chlorides (Cl-)	kg	1.30E-05
(a) Chromium (Cr III, Cr VI)	kg	9.90E-07
(a) Copper (Cu)	kg	1.00E-06
(a) Hydrocarbons (except methane)	kg	4.22E-02
(a) Hydrogen Chloride (HCl)	kg	7.90E-03
(a) Hydrogen Fluoride (HF)	kg	1.60E-04
(a) Lead (Pb)	kg	3.76E-04
(a) Metals (unspecified)	kg	3.80E-04
(a) Methane (CH_4)	kg	2.50E-01
(a) Nickel (Ni)	kg	9.90E-07
(a) Nitrogen Oxides (NO_x as NO_2)	kg	4.24E-01
(a) Nitrous Oxide (N_2O)	kg	7.60E-05
(a) Particulates (unspecified)	kg	3.86E-01
(a) Sulfur Oxides (SO_x as SO_2)	kg	6.58E-01
(a) VOC (Volatile Organic Compounds)	kg	8.53E-03
(a) Zinc (Zn)	kg	1.94E-03
(w) Aluminum	kg	6.50E-03
(w) Ammonium (NH_4+)	kg	1.40E-04
(w) BOD5 (Biochemical Oxygen Demand)	kg	2.18E-03
(w) Chlorine (Cl-)	kg	1.03E-01
(w) COD (Chemical Oxygen Demand)	kg	5.79E-03
(w) Dissolved solids	kg	1.80E-01
(w) Hydrocarbons (CxHy)	kg	2.00E-04
(w) Iron (Fe)	kg	7.00E-03
(w) Nitrogen – tot	kg	2.80E-03
(w) Sodium (Na)	kg	3.10E-02
(w) Sulfate (SO_4--)	kg	4.16E-01
(w) Suspended Matter (unspecified)	kg	5.48E-01
(w) Zinc (Zn++)	kg	1.00E-03

obtained despite major fundamental differences between the LCIA methods applied. For example, the EDIP 97 method does not include a multimedia fate model; the EI 99 method is based on an endpoint approach, whereas the four other methods employ a midpoint approach; and the methods employ different effects models as in the case of USES-LCA and CalTOX. However, on the other hand, there were several procedural difficulties and inconsistencies of results observed.

In general, there were naming convention variations among the five methods investigated that are of importance in the context of metals. The USES-LCA model categorizes the majority of metals by their cationic form. The other four models specify characterization factors for metals in their elemental form. This was determined from supporting documentation references to Chemical Abstract Service (CAS) number as identification nomenclature. In the case copper emissions to air and water, EDIP 97, IMPACT 2002, CalTOX-ETP and EI 99 models specify copper as elemental copper by CAS no. (7440-50-8). In contrast, the USES-LCA model specifies cationic copper (Cu++), CAS no. (15158-11-9). This is not indicative of a simple nomenclature equivalency issue, as the Dutch Handbook method that contains the USES-LCA model recognizes elemental copper, CAS no. (7440-50-8), as an input to abiotic depletion potential (ADP). The cationic/elemental metal discrepancy of nomenclature was problematic for both case studies. For copper tube, emissions to air were gathered as elemental copper (Cu) and emissions to water in its cationic form (Cu++). As shown in Tables 6 and 7 of the Appendix, flow assignments could not be made unambiguously to either air or water.

Universal to nearly all LCA studies, several chemicals identified in the LCIs were not classified by any of the models. Most obvious, and nearly present in all LCIs involving metals, are metals reported as 'unspecified'. For the copper and zinc LCIs were no exception, where unspecified metals represented 50% and 7%, respectively, of the contribution of metals by mass. For LCI inventory items that can be uniquely identified and were not classified, the absence of a characterization factor can be indicative of little or no ecological impact of significance (e.g., air emissions of carbon dioxide and carbon monoxide have no known or little ecological toxicity potential). However, there were instances observed where chemical constituents were recognized by some, but not by all of the models. This was particularly of issue with regards to aluminium impacts observed for the zinc gutter system. Aluminium impacts are characterized by both IMPACT 2002+ and EDIP 97, however, they are not recognized by the other three models. For IMPACT 2002+ aluminium emissions to water were estimated as 90% of the total freshwater ecological impact. For EDIP 97, impact contribution was estimated as slightly over 20%.

As expected for the two case studies, the highest contribution to freshwater ecological toxicity were the primary metals under study, copper and zinc. However, specific inconsis-

Table 4: Zinc gutter system LCIA results by percent contribution

Flow [(a) air; (w) water]	USES-LCA FAETP	CalTOX ETP	EDIP 97 EcoTox	IMPACT 2002+	EI99 HA EcoTox
(a) Lead (Pb)	0.70%	0.12%	3.43%	0.06%	14.49%
(a) Zinc (Zn)	26.72%	98.24%	8.84%	1.57%	85.07%
(w) Aluminium			21.78%	92.79%	
(w) Iron (Fe)			15.95%		
(w) Zinc (Zn++)	71.02%	1.23%	49.94%	5.57%	0.25%
Total Contribution	**98.44%**	**99.58%**	**99.94%**	**99.99%**	**99.81%**

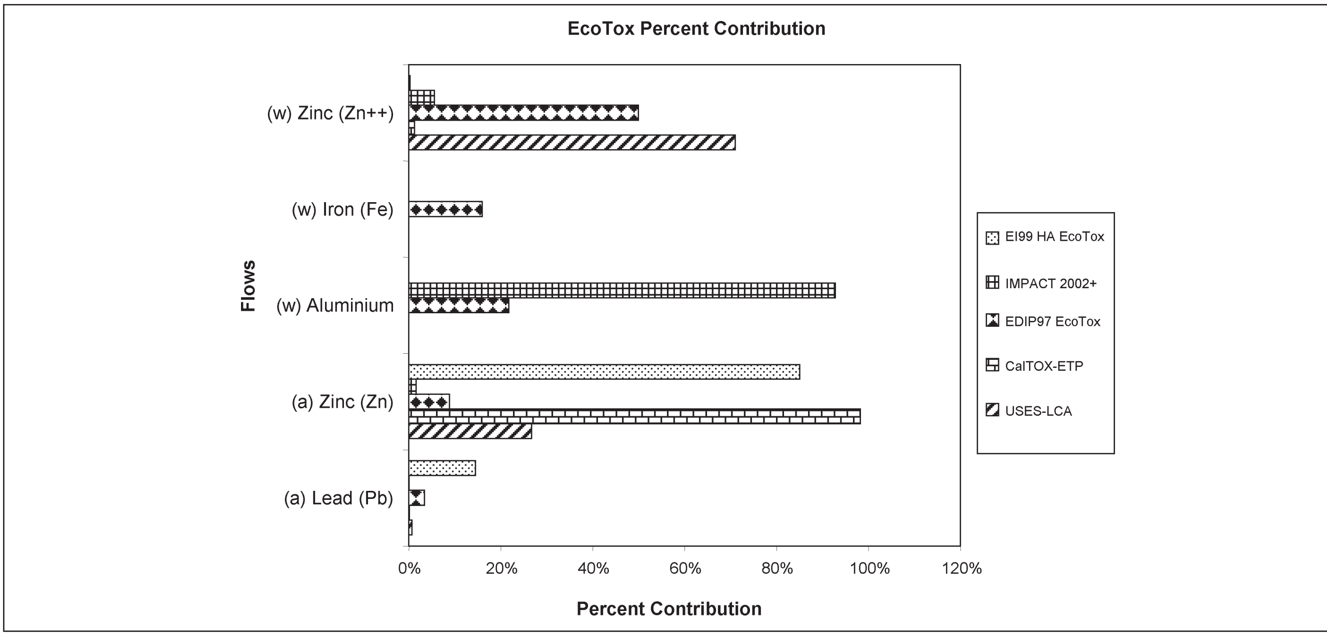

Fig. 4: Zinc gutter system freshwater ecotox results by percent contribution

Table 5: Zinc gutter system LCIA results by percent contribution without lead or iron

Flow [(a) air; (w) water]	USES-LCA FAETP	CalTOX ETP	EDIP 97 EcoTox	IMPACT 2002+	EI99 HA EcoTox
(a) Lead (Pb)	0.70%	0.12%	5.50%	0.83%	14.49%
(a) Zinc (Zn)	26.72%	98.24%	14.20%	21.78%	85.07%
(w) Zinc (Zn++)	71.02%	1.23%	80.20%	77.21%	0.25%
Total Contribution	**98.44%**	**99.58%**	**99.90%**	**99.83%**	**99.81%**

tencies were found for other emissions. Of note, outlier results were observed for water emissions of nickel as characterized by the USES-LCA model and air emissions of cationic zinc (Zn++) and lead (Pb) characterized by the EI 99 model.

For the zinc gutter system study, although zinc was observed as the greatest contributor, contributions varied by media of release. For the EI 99 and the CalTOX-ETP models, contribution to impacts attributed to zinc were related to air emissions exceeded 80% of total. For the USES-LCA model and the EDIP 97 models, water emissions attributed to zinc were 70% and 50% of impact, respectively. The IMPACT 2002 model was dominated by the aluminium contribution to impact, however, impacts attributed to water emissions of zinc were shown to be more than twice the amount of air emissions. Also of note, the EDIP 97 freshwater ecological toxicity model was the only model that contained a characterization factor related to iron. Inconsistency of results for the zinc LCIA remain despite removing lead and iron from the emissions inventory, as shown in **Table 5**.

6 Conclusions

Five LCIA fresh water ecological toxicity models were applied to two case studies involving the production of primarily metal-based products, copper tube and zinc gutter systems. The results obtained from all five of the LCIA methods for the copper tube LCI pointed to the same substance as being the most important – copper. This result was obtained despite major fundamental differences between the LCIA methods applied. However, variations of results were found when examining the freshwater ecological toxicity of zinc gutter systems. In part, this was due to basic differences in the models themselves and differences in coverage (IMPACT 2002+ and EDIP 97 contained characterization factors for aluminium that resulted in 90% and 22% contribution to burden respectively, the other three methods did not). Differences were also observed based on the emissions source compartment specified in the LCI (i.e., air and water). In the case of zinc, air emissions were found to be substantial for CalTOX-ETP and EI 99 models, whereas, water emissions results were found to be of issue for the EDIP 97 and USES-LCA models.

The results demonstrate additional practical issues need to be addressed for ecological toxicity methods to be applied. Issues identified pertain to ambiguities related to classification and inconsistencies found when applying the current characterization methods. Classification issues were due to a lack of specificity in the present nomenclature implemented by the toxicity methods. The use of elemental metals naming conventions are apparently used when referring to cationic species. Further, chemical coverage by the methods is relatively narrow. In the absence of specific classification of a chemical to its ecological toxicity, the practitioner is to assume that there is no impact.

This investigation illustrates the need to proceed with caution when applying ecotox methodologies to life cycle studies that include metals. Until further improvements are made

to the existing ecotox methods, the deficiencies should be clearly communicated as part of LCIA reporting. Business or policy decisions should not without further discussion be made based on the results of the currently available methods for assessing ecotoxicity in LCIA.

7 Outlook

The outlook to remedy the deficiencies in the ecological toxicity methods is promising. Recently, the LCIA Toxic Impacts Task Force of the UNEP/SETAC Life Cycle Initiative formed a subgroup to address the issues specific to ecotoxicity impacts of metals and guide the work towards the establishment of scientifically based characterization factors for metals. Support for this activity has come not only from the broad scientific community, but also from industry as well. Progress is underway to more fully understand the importance of LCIA modelling improvement and to investigate possible options that are based on science and practical to implement.

Although some measure of precision of estimation of potential impact has been observed, such as in the case of copper, accuracy is also a major concern and should be addressed. Further investigation through controlled experimentation is needed, particularly LCIs composed of a variety of inorganics as well as organics constituents.

In practice, flow definition issues can be nontrivial. It is not uncommon for LCI results to contain several mismatches or synonyms to chemical names. Previous efforts to address this issue have been attempted by international and governmental organizations (SETAC Working Group on Data Quality and Data Availability (de Beaufort-Langeveld et al. 2003) and EcoInvent database (Frischnecht et al. 2005) and more recently the mandate by the European Commission Directorate General JRC to develop an European Reference Life Cycle Assessment Data System (ELCD)), however, international consensus has yet to be achieved. Broader aspects of structure and nomenclature are being collectively addressed by the LCI Database Characteristics and Quality Task Force and the LCIA Information Task Force of the UNEP/SETAC Life Cycle. These efforts will bring practical solutions to issues of naming conventions and LCI to LCIA flow assignments.

References

Bare J, Norris G, Pennington D, McKone T (2003): TRACI: The Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts. Journal of Industrial Ecology 6 (3–4) 49–78

Beaufort-Langeveld A de, Bretz R, Hischier R, Huijbregts M, Jean P, Tanner T, Hoof G van (2003): Code of Life Cycle Inventory Practice, Pensacola, FL, Society of Environmental Toxicology and Chemistry (SETAC)

Boustead Consulting, Ltd. (1998): Ecoprofile of primary zinc production. Confidential Report. February 29, pp 149–157

Di Toro D, Allen H, Bergman H, Meyer J, Paquin P (2001): Biotic Ligand Model of the Acute Toxicity of Metals: 1. Technical Basis. Environmental Toxicology and Chemistry 20 (10) 2383–2396

Ecobalance (2000a): Life Cycle Assessment of Copper Tube for the Supply of Water in Residential Structures: Final Report – Peer Review. Ecobalance – Price Waterhouse Coopers, Bethesda, MD

Ecobalance (2000b): Life Cycle Assessment of Copper Tube for the Supply of Water in Residential Structures: Peer Review Methodology Report. Ecobalance – Price Waterhouse Coopers, Bethesda, MD

Eggels P, Giljswijk R van, Ven B van der (2000): Environmental Analysis of zinc gutters and down pipes LCA, directed at Milieurelevante Productinformacie (MRPI). TNO-report TNO-MEP – R 2000/31458, Apeldoorn, NL

EC (European Commission) (1996): Technical guidance documents in support of Directive 93/67/EEC on risk assessment of notified substances and Regulation (EC) No. 1488/94 on risk assessment of existing substances (Parts I, II, III and IV). Luxembourg, L, Office for Official Publications of the European community, CR-48-96-001

Frischknecht R, Jungbluth N, Althaus H-J, Doka G, dones R, Heck T, Hellweg S, Hischier R, Nemecek T, Rebitzer G, Spielmann M (2005): The ecoinvent Database: Overview and Methodological Framework. Int J LCA 10 (1) 3–9

Goedkoop M, Spriensma R (1999): The Eco-Indicator 99: A damage orientated method for life cycle impact assessment, VROM, Ministry of Housing, Spatial Planning, and the Environment, the Hague, The Netherlands

Guinée J, Gorrée M, Heijungs R, Huppes G, Kleijn R, Koning A de, Oers L van, Wegner Sleeswijk A, Suh S, Udo de Haes H, Bruijn H de, Duin R van, Huijbregts M, Lindeijer E, Roorda A, Ven B van der, Weidema BP (2002): Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards, Kluwer Academic Publishers, Dordrecht

Hauschild M, Wenzel H (1998): Environmental Assessment of Products Vol. 2. Chapman & Hall, New York

Heijungs R, Koning A de, Ligthart T, Korenromp R (2004): Improvement of LCA characterization factors and LCA practice for metals. TNO-report R 2004/347. Apeldoorn, The Netherlands <http://www.leidenuniv.nl/interfac/cml/ssp/projects/lcia-metals.html>

Huijbregts M, Thissen U, Guinée, J, Jager T, Meent D van de, Ragas A, Wegener Sleeswijk A, Reijnders L (2000a): Priority assessment of toxic substances in life cycle assessment. Part I: Calculation of toxicity potentials for 181 substances with the nested multi-media fate, exposure and effects model USES-LCA. Chemosphere 41, 541–573

Huijbregts M, Thissen U, Jager T, Meent D van de, Ragas A (2000b): Priority assessment of toxic substances in life cycle assessment. Part II: Assessing parameter uncertainty and human variability in the calculation of toxic potentials. Chemosphere 41, 575–588

Huijbregts M, Guinée J, Reijnders L (2001): Priority assessment of toxic substances in life cycle assessment III: Export of potential impact over time and space. Chemosphere 44, 59–65

Jolliet O, Assies J, Bovy M, Finnveden G, Guinée J, Hauschild M, Heijungs R, Hofstetter P, Potting J, Udo de Haes H, Wrisberg N (1996): Impact assessment of human and ecotoxicity in life cycle assessment. In: Udo de Haes H (ed), Towards a methodology for life cycle impact assessment. Brussels, Society of Environmental Toxicology and Chemistry (SETAC), Pensacola, FL

Jolliet O, Margni M, Charles R, Humbert S, Payet J, Rebitzer G, Rosenbaum R (2003): IMPACT 2002+: A new life cycle impact assessment methodology. Int J LCA 8 (6) 324–330

Jolliet O, Müller-Wenk R, Bare J, Brent A, Goedkoop M, Heijungs R, Itsubo N, Peña C, Potting J, Pennington D, Rebitzer G, and Schenck R (2004): The LCIA midpoint-damage framework of the UNEP/SETAC life cycle initiative. Int J of LCA 9 (6) 394–404

Ligthart T, Aboussouan L, Adams W, Atherton J, Baukloh A, Chapman P, Delbeke K, Dubreuil A, Eikelboom R, Hauschild M, Heijungs R, Jolliet O, Karman C, Koning A de, Korenromp R, Kuyper J, Loos R van der, Meent D van de, Russell A, Jan Saft R, Sap G, Schönnenbeck M, Struijs J, van Tilborg W, Udo de Haes H, Verdonck F (2004): Declaration of Apeldoorn on LCIA of Non-Ferro Metals. Apeldoorn, NL <http://www.uneptie.org/pc/sustain/reports/lcini/Declaration_of_Apeldoorn_final.pdf>

Meyer J, Santore R, Bobbitt J, Debrey L, Boese C, Paquin P, Allen H, Bergman H, Ditoro D (1999): Binding of Nickel and Copper to Fish Gills Predicts Toxicity When Water Hardness Varies, But Free-Ion Activity Does Not. Environ Sci & Technol 33 (6) 913–916

McKone T (2001): Ecological Toxicity Potentials (ETPs) for Substances Released to Air and Surface Waters, Summary Report, unpublished document, submitted to Systems Analysis Branch, Sustainable Technology Division, National Risk Management Research Laboratory, Office of Research and Development, US Environmental Protection Agency, Cincinnati, OH, December

Morel F, Hering J (1993): Principles and Applications of Aquatic Chemistry, Wiley, New York

Pennington D, Margni M, Payet J, Charles R, Jolliet O (2005): Toxicological effect indicators in life cycle assessment (LCA): Comparative versus regulatory risk approaches. Submitted to Human and Ecotoxicological Risk Assessment Journal

Udo de Haes H, Lindeijer E (2002): The Conceptual Structure of Life-Cycle Impact Assessment. In: Udo de Haes H et al. (ed), Life-Cycle Impact Assessment: Striving Towards Best Practice. Society of Environmental Toxicology and Chemistry Press, Pensacola, FL, pp 209–225

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Appendix

Table 6: Copper tube freshwater ecological toxicity classification and characterization results*

LCI Emission [(a) air; (w) water]	Flow Assignment	CAS No.	USES-LCA (kg DCB-equ.)	CalTOX-ETP (kg 2,4 D-eq.)	EDIP 97 (m^3 water or soil)	IMPACT 2002+ (kg triethylene glycol eq.)	EI99 HA EcoTox ($PDF*m^2*a$)
(a) Ammonia (NH_3)	ammonia (NH_3)	7664-41-7				2.01E-02	
(a) Arsenic (As)	cation (AsV)	17428-41-0	3.00E-04				
	elemental (As)	7440-38-2		3.64E-05	2.33E+00	3.32E-01	3.59E-03
(a) Cadmium (Cd)	cation (Cd++)	22537-48-0	6.54E-04				
	elemental (Cd)	7440-43-9		9.72E-06	5.32E+01	9.66E-01	2.18E-02
(a) Chromium (Cr III, Cr VI)	cation (Cr VI)	16065-83-1	4.13E-06				
	elemental (Cr)	7440-47-3		8.60E-06	2.87E-01	1.44E-01	8.88E-03
(a) Cobalt (Co)	cobalt (Co)	7440-48-4	8.44E-05	1.08E-05	5.28E-02	7.14E-02	
(a) Copper (Cu)	cation (Cu++)	15158-11-9	2.73E-02				
	elemental (Cu)	7440-50-8		2.21E-02	3.08E+02	3.62E+02	1.80E-01
(a) Lead (Pb)	lead (Pb)	7439-92-1	4.92E-05	4.92E-06	8.20E+00	8.21E-01	5.21E-02
(a) Mercury (Hg)	cation (Hg++)	14302-87-5	1.61E-05				
	elemental (Hg)	7439-97-6		3.72E-05	2.04E-01	4.00E-02	4.22E-05
(a) Nickel (Ni)	cation (Ni++)	14701-22-5	8.75E-04				
	elemental Ni	7440-02-0		1.81E-04	1.85E-01	2.48E-01	9.87E-03
(a) Silver (Ag)	(a) Silver (Ag)	7440-22-4			1.07E+00	0.00E+00	
(a) VOCs	VOCs		9.01E-06				
(a) Zinc (Zn)	cation (Zn++)	23713-49-7	1.90E-04				
	elemental (Zn)	7440-66-6		4.07E-04	2.14E+00	2.18E+00	3.09E-02
(w) Cadmium (Cd++)	cation (Cd++)	22537-48-0	7.07E-06				
	elemental (Cd)	7440-43-9		1.27E-06	5.46E-01	1.35E-02	2.23E-06
(w) Chromium (Cr III, Cr VI)	cation (Cr VI)	16065-83-1	1.91E-07				
	elemental (Cr)	7440-47-3		1.44E-08	1.84E-02	1.25E-02	1.90E-06
(w) Cobalt (Co I, Co II, Co III)	elemental (Co)	7440-48-4	3.31E-07	5.72E-10	1.94E-04	3.75E-04	
(w) Copper (Cu+, Cu++)	cation (Cu++)	15158-11-9	2.27E-03				
	elemental (Cu)	7440-50-8		9.08E-05	2.45E+01	4.03E+01	2.88E-04
(w) Lead (Pb++, Pb_4+)	cation (Pb++)	14280-50-3	1.20E-06				
	elemental (Pb)	7439-92-1		1.63E-08	2.50E-01	3.30E-02	9.24E-07
(w) Mercury (Hg+, Hg++)	cation (Hg++)	14302-87-5	5.91E-07				
	elemental (Hg)	7439-97-6		4.13E-08	1.38E-03	5.43E-03	6.78E-08
(w) Nickel (Ni++, Ni_3+)	cation (Ni++)	14701-22-5	2.24E-03				
	elemental (Ni)	7440-02-0		2.22E-06	4.62E-01	8.82E-01	9.91E-05
(w) Silver (Ag+)	elemental (Ag)	58594-72-2			1.82E-04	0.00E+00	
(w) Zinc (Zn++)	cation (Zn++)	23713-49-7	7.75E-05				
	elemental (Zn)	7440-66-6		7.77E-07	8.45E-01	1.19E+00	1.38E-05

*Table cells that are crossed out indicate that a flow assignment was not made for the impact assessment model

Table 7: Zinc Gutter freshwater classification and characterization results*

LCI Emissions [(a) air; (w) water]	Flow Assignment	CAS No.	USES-LCA FAETP (kg DCB-equiv.)	CalTOX-ETP (kg 2,4 D-equiv.)	EDIP 97 (m^3 water or soil)	IMPACT 2002+ (kg triethylene glycol equiv.)	EI99 HA EcoTox ($PDF*m^2*a$)
(a) Ammonia (NH_3)	Ammonia (NH_3)	7664-41-7				5.52E-02	
(a) Chromium (Cr III, Cr VI)	cation (CrVI)	18540-29-9	7.61E-06				
	elemental (Cr)	7440-47-3		3.96E-06	1.32E-01	6.64E-02	4.09E-03
(a) Copper (Cu)	cation (Cu++)	15158-11-9	2.22E-04				
	elemental (Cu)	7440-50-8		1.80E-04	2.50E+00	2.94E+00	1.46E-03
(a) Lead (Pb)	elemental (Pb)	7439-92-1	9.02E-04	9.02E-05	1.50E+02	1.51E+01	9.55E-01
(a) Nickel (Ni)	cation (Ni++)	14701-22-5	6.23E-04				
	elemental (Ni)	7440-48-4		1.29E-04	1.32E-01	1.77E-01	7.03E-03
(a) VOCs	VOCs		4.23E-04				
(a) Zinc (Zn)	cation (Zn++)	23713-49-7	3.45E-02				
	elemental (Zn)	7440-66-6		7.37E-02	3.88E+02	3.96E+02	5.61E+00
(w) Aluminum	Aluminum	7429-90-5			9.56E+02	2.34E+04	
(w) Iron (Fe)	Iron (Fe)	7439-89-6			7.00E+02	0.00E+00	
(w) Zinc (Zn++)	cation (Zn++)	23713-49-7	9.17E-02				
	elemental (Zn)	7440-66-6		9.20E-04	4.00E+00	1.40E+03	1.63E-02

*Table cells that are crossed out indicate that a flow assignment was not made for the impact assessment model